MANAGEMENT OF IRRIGATED AGRICULTURE TO INCREASE CARBON STORAGE

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Abstract

Fossil fuel burning at the present rate, will double atmospheric carbon dioxide (CO_2) in this century, raising air temperature 1.5 to 5 °C. Sequestering carbon (C) in soil can reduce atmospheric CO_2 concentration. We measured inorganic and organic C in southern Idaho soils having long term land use histories of native sagebrush vegetation (NSB), irrigated moldboard plowed crops (IMP), irrigated conservation- (chisel) tilled crops (ICT) and irrigated pasture systems (IP). Soil Organic C (SOC) decreased in the order IP>ICT>NSB>IMP. We used our findings to estimate potential amounts of organic C sequestered if irrigated agriculture expanded. If irrigated agricultural land was expanded by10% worldwide and NSB was converted to ICT, 2.5 x 10^9 Mg organic C (4.38 % of the total C emitted in the next 30 yr) could potentially be sequestered in soil. If irrigated agricultural land were expanded by 10% worldwide and NSB were converted to IP, a possible 9.3 x 10^9 Mg organic C (16.32 % of the total C emitted in the next 30 yr) could be sequestered in soil. Irrigated agriculture produces twice the yield compared to non-irrigated land. Irrigation increases soil C relative to native semi arid or arid sites. Since irrigated agriculture produces higher yields, less land area needs to be put into production compared to rainfed agriculture. Altering land use to produce crops on high output irrigated agriculture, while returning less-productive rainfed agricultural land to temperate forest or native grassland, could further reduce atmospheric CO_2. Inorganic carbon increases with irrigation were less consistent and much smaller than SOC. Irrigating these soils also increased microbial biomass and changed microbial diversity.

Introduction

Atmospheric carbon dioxide (CO_2) is expect to double during the next century if fossil fuel burning continues at the present rate, raising air temperature 1.5 to 5 °C (Streck, 2005; Post et al., 2004). Management of vegetation and soils for terrestrial carbon (C) sequestration can remove significant amounts of CO_2 from the atmosphere and store it as C in the organic matter of ecosystems (Boody et al., 2005; Smith, 2005; West and Post, 2002). The amount of C sequestered in soil by land management could be significantly increased. While these rates will offset only a fraction of the emissions from fossil fuels, results from integrated assessment analyses indicate that soil C sequestration could be an important strategy due to its low cost and potential for early deployment within a portfolio of climate change mitigation technologies (Post et al., 2004). Unlike many other technologies to offset fossil fuel emissions, land management changes can be implemented fairly rapidly. Some land management strategies for C sequestration can be expensive to implement because of increased risk and reduced yield and income (Kurkalova, 2005; Post et al., 2004). For soil management to be an effective C sequestration strategy, practices must be cost effective and attractive to land managers and will likely require financial incentives to encourage management changes (Kurkalova, 2005; Post et al., 2004).

Soil is the largest pool of C in the terrestrial environment (Smith, 2005; West and Post 2002). The amount of C present in soil is twice the amount of C in the atmosphere and three times the amount of C in living plants. Therefore, increasing the size of the soil organic C pool could significantly lower atmospheric CO_2 concentration (Wang et al., 1999). The concentration of C in soil is a balance between past C accumulation and loss. Soil C accumulation derives from litter and root input. Losses result from microbial degradation of organic matter and erosion. At equilibrium, the rate and amount of C added to soil via vegetation equals the rate and amount of C lost through organic matter degradation and other losses (Entry and Emmingham, 1998).

Land use changes affect the amount of C stored in soil by altering C inputs and losses. In forest, grassland and wetland ecosystems, conversion of native vegetation to agricultural cropping has resulted in substantial C transfer to the atmosphere due to loss of climax vegetation and to a lower equilibrium C concentration in soil (Wang et al., 1999). In arid and semi-arid environments plant survival and growth is limited by available water. Irrigation is required for economically viable crop production; it also can increase C input to soils via increased litter and root production.

Materials and Methods

Site descriptions

The study area is located on the Snake River Plain, between 42° 30' 00" and 43° 30' 00" N. and 114° 20' 00" and 116° 30' 00" W. The sites occur across an elevation gradient ranging from 860 m to 1300 m. The area is classified as a temperate semi-desert ecosystem (Bailey, 1998). The climate has cool, moist winters and hot, dry summers with annual precipitation ranging 175 to 305 mm, two-thirds of which occurs during October through March (Collett, 1982). Annual mean temperature ranges from 9 to 10 ° C. Soils are typically well-drained loams derived from loess deposits overlying basalt. Vegetation throughout the area was historically dominated by basin big sagebrush (*Artemisia tridentata* var. *tridentata* Nutt.), Wyoming big sagebrush (*Artemisia tridentata* var. *wyomingensis* Nutt.), and perennial bunch grasses.

Experimental design

The experiment was arranged in a completely randomized design (Kirk, 1982). Soil samples were taken from: 1) three sites supporting native sagebrush vegetation (NSB) located near agricultural land in Southern Idaho; 2) three sites that were formerly crop land and converted to and maintained as irrigated pasture (IP) for the past 30 years; 3) three sites that were irrigated crop land and have been managed with conservation tillage (ICT) for the past 8 years; and 4) three irrigated agricultural crop lands in moldboard plowing systems (IMP) that were each growing a) alfalfa (*Medicago sativa* L.), b) wheat (*Triticum aestivum* L.), c) potato (*Solanum tuberosum* L.) and d) beans (*Phaseolus vulgaris* L.). There were four treatments (NSB, IMP, ICT, and IP) x three sites per treatment x five cores taken within each treatment at each site (reps) x four soil depths (0-5 cm, 5-15 cm, 15-30 and 30-100 cm) . We took a total of 240 samples.

Site Descriptions

Native sagebrush sites were vegetated with native steppe vegetation and a low composition of exotic annual grasses. Sites were chosen for this study based on a history of no livestock grazing (BLM, Bruneau Resource Area, unpublished data). All study sites had 5-10 % slope and were on areas that supported basin big sagebrush or Wyoming big sagebrush or communities. Soil was classified as a fine, montmorillonitic, mesic Xerollic Haplargid on the Brown's Creek site, a coarse-loamy, mixed non-acid, mesic Xeric Torriorthents on the Simco site and a loamy, mixed, mesic lithic Xerollic Camborthids on the Kuna Butte site (Collett, 1982).

Three irrigated pastures were selected that were formerly crop land and converted to and maintained as irrigated pasture (IP) for the past 30 years. The Buhl site was vegetated with a Kentucky bluegrass (*Poa pratensis* L.)–orchardgrass (*Dactylis glomerata* L.) sward on a Rakane-Blacknest soil complex, fine-loamy, mixed, mesic Xerollic Durargids soil. The Gooding site was vegetated with smooth brome (*Bromus inermis* Leyss.)-orchardgrass on a Paulville-Idow soil complex, fine-loamy, mixed, mesic Xerollic Haplargid soil. The Kimberly site was vegetated with smooth brome-orchardgrass pasture on a Portneuf soil, coarse-silty, mixed, superactive, mesic Durinodic Xeric Haplocalcid. Grazing on the pastures were 10-12 animal unit months yr^{-1}.

Three sites with fields rotating among alfalfa (*Medicago sativa* L.), wheat (*Triticum aestivum* L.), potato (*Solanum tuberosum* L.) and beans (*Phaseolus vulgaris* L.) were sampled. All sites were located on fields managed by USDA Agricultural Research Service's Northwest Irrigation and Soils Research Laboratory or the University of Idaho. Soil on all sites was classified as a coarse-silty, mixed, superactive, mesic Durinodic Xeric Haplocalcid, with 0.1-0.21 g/g clay and 0.6-0.75 g/g silt, and organic matter of approximately 13 g kg^{-1}. The soil has a pH of 7.6-8.0. Slope on these sites ranges from 1.0 to 3.0%.

Sampling Procedures

Soil cores were taken from each site during winter (January), spring (April), summer (August), and autumn (November) in 1999. We sampled the top 1 m of soil each season (winter, spring, summer and autumn) to determine if the amount of C in soil would be affected by irrigation, tillage and vegetation. Sampling locations were randomly chosen at each site or field. Separate 10 cm diameter replicate cores were randomly taken and partitioned into 0-5 cm, 5-15 cm, 15-30 and 30-100 cm depths. Roots greater than 1.0 cm diameter were measured separately. Carbon in above-ground vegetation was estimated by measuring the amount of material in 10 separate 1.0 m^2 areas in each site or field (Entry and Emmingham, 1998).

Carbon in Soil and Above-ground Vegetation

Concentration of organic C in each sample of mineral soil was determined by the Walkley-Black procedure and loss on ignition (Nelson and Sommers, 1996). The amount of C per ha^{-1} of the 0-100 cm of mineral soil was calculated assuming 0.44 g C g^{-1} organic matter with correction for soil bulk density. Ten separate 10 cm diameter soil cores were taken to a 1.0 m depth, divided into 0-5 cm, 5-15 cm, 15-30 and 30-100 cm depths to determine bulk density. Bulk density was measured by dividing by the oven dry weight after drying at 105 °C for 48 hours by the volume of the sample (Blake and Hartage, 1982). Above-ground vegetation was collected and separated into sage, grass, forbs, herbs and duff. Above-ground material was dried at 80 °C for 48 h, weighed and ground to pass a 1 mm opening. Carbon in above-ground vegetation was determined by loss on ignition (Nelson and Sommers, 1996). The amount of

C in the above ground material was assumed to contain 0.44 g C g^{-1} organic matter on an ash free basis (Nelson and Sommers, 1996). Calculations to estimate C stored in soils in the Western United States and worldwide were based on the Walkley Black procedure.

Calculations
Concentration of organic C determined by the Walkley-Black procedure was converted, using bulk density measurements, to a meter square basis to a depth of 1 meter. Organic C in kg m^{-2} was converted to Mg C ha^{-1} by multiplying by 10,000 (land area) and dividing by 1,000 (C weight), which is a 1:10 conversion. We divided the resulting number by 10 to account for a 10 % conversion of one treatment (land area). The amounts of C sequestered in the Pacific Northwestern United States, the 11 western states in the United States, and worldwide were estimated by multiplying Mg C ha^{-1} by the number of hectares of irrigated land in each region. There are 9,055,979 ha of irrigated crop land in the Pacific Northwest, 24,322,029 ha in the Western U.S. and 260, 000,000 ha worldwide (Bucks et al., 1990; Tribe, 1994; Howell, 2000). The carbon sequestered worldwide (C_S) relative to the amount of C projected to be emitted worldwide during the next 30 yr (C_{EW}) was calculated by dividing the Mg C sequestered in each treatment x area, by the total projected worldwide release of CO_2 - C during the next 30 years (5.7 x 10^{10} Mg C) multiplied by 100.

Statistical analysis
All data were subjected to a one way vegetation-type analysis of variance (ANOVA) for a completely randomized design (Snedecor and Cochran, 1980; Kirk, 1982). Residuals were normally distributed with constant variance. Statistical Analysis Software programs were used to conduct the analysis of variance. Significance of treatment means was determined at $P<0.05$ with the Least Square Means test.

Results and Discussion
Site Specific Findings
ANOVA showed that soil bulk density, soil C, site C, net C in soil, net site C in site x vegetation interactions are not significant at $P \leq 0.05$. Therefore, results are discussed with respect to vegetation differences (Snedecor and Cochran 1980; Kirk 1982). Soil C was greater in the NSB 0-5 cm soil depth than the 5-15, 15-30 and 30-100 cm depths and all other soils (Table 1.). Soil C was greater in the in the 0-5 cm, 5-15 cm and 15-30 cm depths in the IP than the IMP or ICT treatments. Organic C contained in above-ground vegetation was greater on NSB sites than IP; however, IP biomass was removed by grazing. Crops were not considered as permanent vegetation. Prior to adjustment for agricultural CO_2 emissions, "gross" soil C and C on site was greatest to least in the order IP>ICT>IMP>NSB (Table 2). IP management emits less C to the atmosphere than IMP- or ICT- agriculture because fewer fertilizer and operational inputs are necessary. After adjustment for agricultural CO_2 emissions, "net" C in soils was greatest to least in the order IP>ICT>NSB>IMP.

We estimated that if NSB sites were converted to IMP a net release of 0.15 kg C m^{-2} over 30 years would occur. We estimated a net sequestration of 0.80 kg C m^{-2} over 30 years if NSB sites were converted to ICT, and if converted to IP one could expect a net sequestration of 3.56 kg C m^{-2} over 30 years. We estimated that if IMP was converted to ICT a net sequestration of 0.95 kg C m^{-2} over 30 years would result from the conversion. If IMP land was converted to IP an estimated net sequestration of 3.71 kg C m^{-2} over 30 years would result from that conversion.

Table 1. Bulk density, Walkley Black C and loss on ignition C (LAI-C) in soils growing native sagebrush, irrigated moldboard plowed cropland, irrigated conservation tilled cropland and irrigated pastures in Southern Idaho.

Treatment	Soil Depth	Bulk Density	WBC†	LAI-C
		-- Mg m^{-3} --	--------- g C kg^{-1} soil----------	
Native sagebrush	0-5	0.97 b	127 a	101 a
	5-15	1.28 ab	47 c	41 c
	15-30	1.34 a	54 c	51 c
	30-100	1.36 a	40 c	35 c
Irrigated moldboard plow crop	0-30 ‡	1.28 ab	78 b	69 b
	30-100	1.37 a	60 bc	51 bc
Irrigated conservation tilled crop	0-15 ‡	1.38 a	89 b	76 b
	15-30 ‡	1.38 a	69 bc	68 bc
	30-100	1.37 a	37 c	31 c
Irrigated pasture	0-30 ‡	1.33 a	85 b	79 b
	30-100	1.40 a	43 c	38 c

† In each column, values followed by the same letter are not significantly different as determined by the Least Square Means Test (P 0.05), n = 30.

‡ Statistical comparisons in the ANOVA showed that soil bulk density with respect to soil depth were not significant at $P \leq 0.05$. Therefore, data were combined (Snedecor and Cochran 1980; Kirk 1982).

Table 2. Organic carbon in: soils[1], above ground biomass[2] and on sites at present[3], carbon emitted during agricultural operations[4], net organic carbon in soils[5] and net carbon gain on sites[6].†

	Carbon Present				Net Carbon Gain	
Vegetation	Soil[1] ‡	Above ground[2]	Site[3]	Carbon emitted[4] §	Soil[5] ‡	Site[6]
	kg C m^{-2}					
Native sagebrush	5.91 c	0.42 a	6.34 c	0.00 d	5.91 c	6.34 c
Irrigated moldboard plow crops	7.29 b	0.00 c	7.29 b	1.10 a	6.19 b	6.19 c
Irrigated conservation till crops	8.01 b	0.00 c	8.01 b	0.87 b	7.14 b	7.14 b
Irrigated pasture	10.14 a	0.05 b	10.19 a	0.29 b	9.85 a	9.90 a

† In each column, values followed by the same letter are not significantly different as determined by the Least Square Means Test (*P* # 0.05), n = 30.
‡ Values of organic C stored in soils are based on the Walkley Black procedure.
¶ Carbon in soils, above ground vegetation and on the sites at the present time.
§ Estimated carbon emitted in production of fertilizer, fuel consumption in farm operations and via irrigation water over a 30-year period.

Regional and Global Implications

Strategic changes in land management and agricultural practices have great potential to sequester C. In most cases converting selected land managed as IMP to ICT or IP can be implemented with modest economic impact to landowners and pose relatively few socioeconomic issues. Estimating the potential for C sequestration with precision in terrestrial ecosystems is difficult because the dynamics that control C flow among plants, soils and the atmosphere are poorly understood. Storage of C in below ground systems is the best long-term option in terrestrial ecosystems because C in soils has a longer residence time than in most plant biomass.

We used the insight from our measurements to pose additional strategic scenarios based on the carbon pool impact we measured and on existing knowledge about irrigated yield advantages and relative ecosystem carbon storage. Using the values obtained in southern Idaho, we estimated C storage gains if: 1) 10% of irrigated land now in IMP agriculture was converted back to NSB, 2) all land presently in IMP was converted to ICT and 3) 10 % of land in irrigated IMP was converted to IP. Since increased agricultural production will be necessary to feed an increasing population, it is impractical to suggest that a large portion of land in IMP can be converted to IP.

The reported amounts of C stored in native sagebrush vegetation and in irrigated agriculture are similar throughout the U.S. and worldwide (Bowman et al., 1999; Amthor and Huston, 1998). These data were used to calculate potential C storage for irrigated agriculture worldwide over 30 years. If land currently in IMP is converted to NSB we estimate a gain of 1.5 Mg C ha^{-1}. If IMP land is converted back to native sagebrush (NSB), 5.4 $x10^{7}$ Mg C (0.10% of the total C emitted in the next 30 yr) could be sequestered in the next 30 years (Table 3). Converting irrigated land back to native sagebrush (NSB) is not practical since the land is needed for food production and would not appreciably increase C sequestration.

Little irrigated crop land is managed with conservation tillage. We estimate an increase of 9.5 Mg C ha^{-1} over 30 years if the land presently managed with IMP were converted to ICT. If the IMP land were converted to ICT, $2.5x10^{9}$ Mg C (4.38% of the total C emitted in the next 30 yr) could be sequestered in the next 30 years. A shift of 10 % of current IMP land to ICT is a reasonable conservation practice goal. Similarly, we cannot expect all land presently in IMP to be converted to IP, but a 10 % conversion is feasible. If we predict a storage increase of 37.1 Mg C ha^{-1} for IMP converted to IP and assume 10 % of IMP land is converted to IP, an estimated $9.6x10^{8}$ Mg C (1.68% of the total C emitted in the next 30 yr) could be sequestered if irrigated land presently managed as IMP were converted to ICT (Table 3). If irrigated agriculture is expanded due to development of new water resources or an increase in water use efficiency, one could expect a gain in C sequestration. If NSB were converted to ICT, 8.0 g Mg C ha^{-1} could be sequestered. Predicting a storage increase of 8.0 Mg C ha^{-1} for NSB converted to ICT and assuming 10 % expansion of irrigated agriculture worldwide, an estimated $2.1x10^{8}$ Mg C (0.37% of the total C emitted in the next 30 yr) could be sequestered. If NSB were converted to IP, then 35.6 g C ha^{-1} could be sequestered. Predicting a storage increase of 35.6 Mg C ha^{-1} for NSB converted to IP and assuming 10% expansion of irrigated agriculture, an estimated $9.3x10^{9}$ Mg C (16.32% of the total C emitted in the next 30 yr) could be sequestered.

Table 3. Potential organic C gain by conversion of irrigated lands currently in moldboard plowing systems to conservation tillage, conversion of native sagebrush to irrigated conservation tillage, conversion of native sagebrush to irrigated pasture and conversion of 10% of irrigated lands currently in moldboard plowing systems to irrigated pasture over the next 30 years.

Vegetation Conversion	C gained from a 10 % conversion	Worldwide ‡	
	--Mg C ha^{-1}--	Mg C	%C_S/C_{EW} §
Irrigated moldboard plow (IMP) to native sagebrush (NSB)	1.5	5.5×10^{7}	0.10
Irrigated moldboard plow (IMP) to irrigated conservation tillage (ICT)†	9.5	2.5×10^{9}	4.38
Native sagebrush (NSB) to irrigated conservation tillage (ICT)	8.0	2.1×10^{8}	0.37
Native sagebrush (NSB) to irrigated pasture (IP)	35.6	9.3×10^{9}	16.32
10% of irrigated moldboard plow (IMP) to irrigated pasture (IP) †	37.1	9.6×10^{8}	1.68

† Estimated C gain from 100% conversation of moldboard plow to conservation tillage and 10% conversion of moldboard plow agriculture to irrigated pasture.

‡ Land area in irrigated cropland worldwide (260,000,000 ha).

§ %C_S/C_{EW} = carbon sequestered (C_S) divided by the amount of C projected to be emitted worldwide during the next 30 yr, which is 5.7×10^{10} Mg C (C_{EW}) multiplied by 100.

Since the earth releases 1.9 x 10^9 Mg C yr^{-1} (Schlesinger, 1999; Amthor and Huston, 1998), the conversion of 10% IMP to IP, resulting in a possible 9.6x10^8 Mg C sequestered over 30 yrs, may be small. The amount of irrigated agriculture can likely be increased 10% solely through increases in irrigation efficiency and waste water reuse (Howell, 2000), meanwhile a number of large new irrigation developments are underway in several regions of the world after a 20 yr hiatus of new development. Using a conversion basis of 1 unit of irrigated agriculture to return 1 unit of rainfed agricultural land to native forest, if irrigated agriculture were expanded 10% (meaning that an additional 2.6 x 10^7 ha of arid or semiarid land were irrigated) and the equal amount of land being managed as rainfed agricultural land were converted to native forest, there is potential to sequester 1.5x10^9 Mg C (2.6% of the total C emitted in the next 30 yr) worldwide (Table 4). If the rainfed agricultural land were converted to native grassland, there is a potential to sequester 3.4x10^9 Mg C (5.9% of the total C emitted in the next 30 yr) worldwide.

Table 4. Potential carbon transfer by converting an equal amount (10%) of rainfed moldboard plow land back to native forest or grassland on a basis of 1 unit of irrigated rainfed agricultural land to 1 unit of native forest or grassland and conversion of equal amount (10%) of rainfed moldboard plow land back to native forest or grassland on the basis of 1 unit of irrigated rainfed agricultural land to 2 units of native forest or grassland.

Conversion of Vegetation	C Stored From Conversion	Worldwide	
	Mg C ha^{-1}	Mg C	$\%C_S/C_{EW}$ †
Rainfed moldboard plow to native forest on a 1 unit:1 unit basis	5.6	1.5 x 10^9	2.63
Rainfed moldboard plow to native grassland on a 1 unit:1 unit basis	13.0	3.4 x 10^9	5.96
Rainfed moldboard plow to native forest on a 2 unit:1 unit basis	5.6	3.0 x 10^9	5.26
Rainfed moldboard plow to native grassland on a 2 unit:1 unit basis	13.0	6.8 x 10^9	11.93

The preceding scenarios assume a one to one area trade off between irrigated and non-irrigated land. However, irrigated agricultural land, on average, produces twice the crop yield of rainfed agricultural land (Bucks et al., 1990; Howell, 2000). If irrigated agriculture were expanded 10%, each hectare of new irrigated land could produce the same crop yield as 2 ha of rainfed land (Bucks et al., 1990; Tribe, 1994; Howell, 2000). Under this scenario, 3.0×10^9 Mg C (5.26% of the total C emitted in the next 30 yr) worldwide could be sequestered in soils (Table 4). If converted to native grassland in this 2:1 conversion scenario, there is a potential to sequester 6.8×10^9 Mg C (11.93% of the total C emitted in the next 30 yr) worldwide could be sequestered in soil. If highly erosive rainfed lands were selected, or if rainfed lands urgently needed for habitat restoration were targeted for such conversions, significant additional erosion, water quality and habitat benefits could also result.

Since native desert or semi-desert has relatively little ecosystem C compared to forest, grassland or wetland ecosystems (Houghton et al., 1999; Amthor and Huston, 1998), converting irrigated moldboard plow crop land back to desert or semi-desert could result in a soil C gain of only 0.15 kg C ha^2 equaling a worldwide sequestration of only 5.46×10^7 Mg C 30 yr^{-1} (0.10% of the total C emitted in the next 30 yr). This modest C accumulation in soil would be tempered by the fact that the conversion would require substantial policy incentives and decades to implement. Substantially more C may be sequestered by selectively returning rainfed agricultural land derived from forest, grassland or wetlands back to native vegetation. Tropical and temperate forests typically contain from 10-12 kg C m^{-2}; grasslands contain from 18-20 kg C m^{-2}; and wetland ecosystems contain from 60-70 kg C $^{-2}$. Whereas arid and semi-arid lands contain 5 to 7 kg C m^{-2} (Schimel et al., 2000; Houghton et al., 1999;). Since nearly a third of the yield and nearly half of the value of crops in the U.S. are produced on irrigated lands predominantly in arid or semi-arid climatic zones (Bucks et al., 1990; Tribe, 1994; Howell, 2000), a strong strategic rationale can be made for expanding irrigated agriculture in these areas for both crop production and C sequestration, if accompanied by selective return of rainfed agricultural land derived from forest, grassland or wetlands back to native vegetation.

We also analyzed our samples for irrigation effects on soil inorganic carbon contents. In many irrigated areas of the world, irrigation water is high in carbonates. Irrigation and evaporatranspiration could conceivably result in carbon enrichment of soils via carbonate precipitation. Our data was presented in detail by Entry et al. (2004a). Briefly the impact of irrigation on soil inorganic carbon was much smaller and less consistent than for SOC. The most interesting trend was a slight increase of inorganic carbon under IMP, which was thought to possibly be related to increased soil evaporation with conventional tillage. In general, the impact of irrigated agriculture on inorganic C sequestration in soil as a management practice to offset atmospheric CO_2 is small relative to the impact of fossil fuel burning. Thus, inorganic C sequestered through irrigated agricultural management has a much smaller impact on global C compared to the changes gained through organic C sequestration (Suarez, 2000).

Our studies considered the impact of irrigation on microbial communities. A brief summary of findings is presented here. Readers are referred to the original publication (Entry et al., 2004b) for experimental details and a more thorough interpretation of findings. Briefly, active bacteria and fungal biomass had a strong positive correlation with soil organic C concentrations (Entry et al., 2004b). Increasing soil C increased active bacterial and fungal biomass both during summer months when irrigation water was applied and crop plants were actively growing and during winter when IP, ICT and IMP were not supporting actively growing vegetation.

Entry et al., (2004b) reported that consistent trends in fatty acid methyl ester (FAME) profiles were observed among NSB, IP, ICT and IMP at three separate sampling dates. Mills et al. (2006; 2004) calculated bacterial diversity and evenness indices for four hypervariable regions of the 16S rDNA in these treatments in these same native sagebrush soils (NSB), irrigated conservation tilled soils (ICT), irrigated pasture soils (IP) and irrigated moldboard plowed soils (IMP). The evenness indices better reflected the differences in the soils and management practices based both on depth (i.e, influence of root zone depth and rhizosphere communities) and the associated environmental disturbances (i.e, wind disturbance of top soils) or management practices (i.e., IMP, ICT and IMP) than the diversity indices.

In general, irrigating these arid soils both increased C storage while increasing microbial biomass and changing microbial diversity. In February 2001, active fungal, bacterial and microbial biomass was greater in IP soils than all other soils. Active fungal, bacterial and microbial biomass was least in ICT soils at the 15-30-cm depth than all other soils. In August 2001, active bacterial biomass was greater in IMP soils than IP, ICT, and NSB soils. Active fungal biomass was greater in IP soils than all other soils. Whole-soil-fatty acid profiles differed among management regimes and sampling dates and, to a lesser extent, with soil depth. FAME profiles from the NSB soils were distinct from the agricultural treatments and contained greater amounts of total fatty acids than the other treatments. The IMP and ICT soils yielded fatty acid profiles that were similar to each other, although those at the 15-30 cm depth were distinct from all other treatment-depth combinations. The IP FAME profiles suggest that arbuscular mycorrhizal fungi are more common in these soils than soils from the other treatments. Differences in carbon substrate utilization patterns (BIOLOG) among treatments were more variable and less pronounced than FAME results.

While not a carbon sequestration issue per se, another advantage of building scenarios to reduce global warming around irrigation deserves mention in this context. Irrigation, unlike most rainfed agriculture, results in significant immediate reduction in atmospheric temperature through evapotranspirative cooling that would not occur in its absence. This is particularly true where irrigation is developed on arid or semi-arid land. The magnitude, extent and importance of this immediate cooling effect has recently been recognized (Lobell et al., 2006a,b); in scenarios of doubled atmospheric CO_2, irrigation can provide localized cooling of up to 8 °C and lower mean temperatures by 1 °C in irrigated regions. It is noteworthy that carbon trading schemes have been developed based on eventual future impacts of the indirect and tentative relationship of carbon sequestration to climate management. It is worth contemplating whether perhaps a stronger logic exists to promote irrigation development, via strategic policy and incentives to reward the vast atmospheric cooling service already provided via irrigation of the world's arid and semi-arid lands.

Factors Affecting Interpretation

Grazing affects the quantity and chemical composition of soil organic matter and the distribution of C in the soil profile (Schuman et al., 1999; Frank et al., 1995). Schuman et al. (1999) and Frank et al. (1995) found that grazing often increases the concentration of soil C. Ecosystems co-evolved with herbivores. The fact that C storage in range and grassland ecosystems may be unaffected, but usually is increased with light grazing, suggests that grazing is an important part of long-term sustainability of these ecosystems (Schuman et al., 1999).

We recognize that the values for potential C gain in our study are estimates. To obtain a more precise estimate of potential C sequestration from management conversions on a worldwide basis it would be necessary to investigate the potential C accumulated in soils in many different vegetation types. Use of these data from Idaho provides a suggestion of the potential magnitude these kinds of management shifts could have on carbon management on a larger scale. Our estimated values for C gain may actually be conservative due to improving land management methods and improving irrigation technology. The C trends that we monitored were the end result of management that predated new technology now available that would have prevented much of the erosion and loss of soil C on our monitored irrigation sites (Sojka et al., 2006, 2007). Most irrigated cropping world-wide uses surface irrigation, with runoff resulting in some transport offsite of C via erosion with sediment and dissolved C in water. Additional C that may be sequestered resulting from improved long-term inputs of technology needs to be determined to more accurately predict potential C gains by irrigated agriculture in the future. Our estimates made no attempt to adjust C budgets for loss of C due to erosion. Because great improvements in controlling irrigation-induced erosion have occurred in recent years, it is likely that our C storage estimates for irrigated agriculture are conservative.

Conclusions

When data are averaged within the top 30 cm of soil for native sage brush sites that were converted to irrigated conservation tillage (ICT) or irrigated pasture (IP), soil organic C increased (Entry et al., 2002). If crops were produced via high output irrigated agriculture, while less-productive rainfed agricultural land were returned to temperate forest or native grassland, there could be substantial reductions in atmospheric CO_2. Our data parallel those of DeClerk and Singer (2003) who consistently saw increases in soil organic carbon from 125 irrigated California sites with sixty-year histories of irrigation. Policy makers and agricultural research infrastructure should recognize the enormous potential benefit of land and water management strategies, polices and incentives that could expand arid zone irrigated agriculture as a means of efficient food and fiber production along with substantial C sequestration potential. Such an expansion would have to be accompanied by renewed efforts in water supply development. Irrigation water resource development has resumed at a modest pace worldwide since the 1990s in countries where political instability and/or weak economies had previously prevented such efforts. Also, Howell (2000) indicated the potential for increasing the extent of irrigation via efficiency improvements and waste water use. Recognition of irrigation's potential C benefits should provide further incentive to fund research and pursue management strategies that are possible without sacrificing production and that could increase restoration of native ecosystems, reduce erosion and improve water quality through appropriate targeting of the strategy.

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